

## 16.6.2 Development of Recycling Treatment Technology for Organic Wastes for Controlling CH<sub>4</sub> and N<sub>2</sub>O Emissions (Final Report)

**Contact persons:** Yuhei INAMORI  
Head, International Water Environment Renovation Research Team  
Division of Regional Environment  
National Institute for Environmental Studies,  
16-2 Onogawa, Tsukuba, Ibaraki, 3050053, Japan  
Tel: +81-298-50-2496  
Fax: +81-298-50-2570  
Email: inamori@nies.go.jp

**Total Budget for FY 1998-1999:** 4,919,000 Yen (FY 1999: 2,414,000 Yen)

### ABSTRACT

Emissions of N<sub>2</sub>O and CH<sub>4</sub> from aerated composting system were investigated using small-scale simulated reactors. The results show an emission peak of N<sub>2</sub>O at the beginning of composting in all experimental runs. Nitrous oxide level in the exhausted gas is proportion to the application amount of food waste. After 2 days, N<sub>2</sub>O concentration at air outlet decreases to 0.53 ul·L<sup>-1</sup> in average, near to the background level in the atmosphere (0.45 ul·L<sup>-1</sup>). Addition of compost increased N<sub>2</sub>O emission not only at the beginning of composting, but also during the later period and resulted in two-peak emission curves. Good correlation was observed between N<sub>2</sub>O concentration at the air outlet and NO<sub>2</sub><sup>-</sup> concentration in waste, indicating a generation pathway for N<sub>2</sub>O from NO<sub>2</sub><sup>-</sup> to N<sub>2</sub>O. The phenomenon was further certified by an external addition of nitrite into waste mixture. With the nitrite shock, extremely high N<sub>2</sub>O generation was observed in both compost-applied and non-compost treatments, suggesting the existence of same mechanism for N<sub>2</sub>O production. Nitrous oxide mainly results from the accumulation of nitrite in waste mixture. In compost-applied treatments obvious nitrite accumulation was detected whereas in non-compost treatments most nitrogen was lost by ammonia volatilization and no clear nitrite accumulation was found. Therefore, to reduce N<sub>2</sub>O production, nitrite accumulation should be avoided.

Addition pattern of food waste may also affect N<sub>2</sub>O production. Daily-addition and multi-addition of food waste result into lower N<sub>2</sub>O emission than single addition. Involvement of more carbon source prolongs the ammonification period and postpones N<sub>2</sub>O generation.

Methane was only detected in treatments applied with compost, the high emission of it illustrate the involvement of anoxic/anaerobic microorganisms with the addition of compost. The result suggests the existence of anoxic or anaerobic microsite inside the waste particles even though aeration was employed during composting process.

**Keywords:** N<sub>2</sub>O emission, Methane emission, Food waste, Composting, greenhouse gases

### 1. INTRODUCTION

In recent years, more and more concern was laid on the rapid increases of N<sub>2</sub>O and CH<sub>4</sub> concentrations in stratosphere because they were commonly known to be associated with global warming effects. Lots of studies have so far been conducted to investigate the sources and sinks of

$\text{N}_2\text{O}$  and  $\text{CH}_4$  (Inamori et al., 1991), as well as the mechanism and controlling factors of the emissions (Williams et al., 1992; Wu et al., 1995; Paul et al., 1993). In the past decade, more efforts were taken to develop effective technologies to reduce their emissions (Tanaka et al., 1994). Nevertheless, it should be noticed that most of these studies were carried out in agricultural system, which was commonly recognized as the largest anthropogenic contributor to the global greenhouse gas budget. Waste management was also realized as one of the sources for greenhouse gases, however; the attention paid on it was far from intensive. Treatment of solid waste and wastewater, in which biodecomposition is usually the main mechanism, are reported to produce substantial amount of methane as well as carbon dioxide, even an aerobic condition is employed.  $\text{N}_2\text{O}$  emissions from various waste treatment facilities, esp. wastewater treatment (Frette et al., 1997), incineration (Tanikawa et al., 1995) and manure composting (Czepiel et al., 1996; Mahimairaja et al., 1995), are also frequently documented in recent years. Nevertheless, our knowledge is still limited regarding to the generation mechanisms of these gases, especially  $\text{N}_2\text{O}$ . It is still difficult to calculate the contribution of various waste management facilities to the global greenhouse gases pond is still unknown.

Food waste comprises an important part of domestic refuses. It is usually treated and disposed by landfill, incineration and composting. All these three techniques were found to generate greenhouse gases. Biogenic production of methane from landfills and composting are well understood (Börjesson and Svensson, 1997; Boltze and de Freitas, 1997; Liu and Kunz, 1981; Åkesson and Nilsson, 1998; Kjeldsen and Fischer, 1995; Wang et al., 1997) and are estimated to account for 4 to 15 percent of the total global budget (Tsujimoto et al., 1994). However, little is known about  $\text{N}_2\text{O}$  emissions from the bio-treatment of solid wastes. Tsujimoto et al. (1994) reported  $\text{N}_2\text{O}$  emission from solid waste disposal sites in Osaka. Czepiel et al. (1996) measured  $\text{N}_2\text{O}$  emission from a composted pile of organic wastes and indicated that  $\text{N}_2\text{O}$  flux was functions of the compost age, pile depth, temperature and water filled pore space.

Aerated composting was thought to be a suitable alternative for treatment of organic waste in lieu of landfill and incineration with the consideration of greenhouse gas emission reduction. The high efficiency of aerated composting in degrading organic carbon has been well addressed. However, the gas emissions from such an artificial environment is not known. The purpose of this study was to quantify and characterize the emissions of  $\text{N}_2\text{O}$  and  $\text{CH}_4$  to atmosphere from aerated composting, optimize the operation condition to minimize the emissions of  $\text{N}_2\text{O}$  and  $\text{CH}_4$ .

## 2. MATERIALS AND METHODS

### 2.1 *Experimental apparatus*

Reactors of 18 liters in horizontal cylinder shape were employed in all the experiments. As shown in Fig. 1, waste and amendments were well mixed in the reactors by the slow stirring (0.8 rpm/min) of the motor. Air was imported into the reactor through a perforated plate at the bottom of reactors to keep the aerobic environment. Unless specifically indicated, a same aeration rate of  $0.7\text{L}\cdot\text{min}^{-1}$  on base of 1kg of dry food waste was used. In order to keep the ordinary running during winter, the reactors were put inside thermal-proof chambers. At the same time, two electric heaters were installed, one for the establishment of a warm ambience in the chambers and the other for heating of the aeration gas. A trough was designed below the perforated plate to collect and recycle the possible leachate during the composting process.

The experimental apparatus was placed on a balance so as to monitor the weight changes in the course of operation. Temperature in the reactor was automatically determined by the sensor buried the waste mixture.

### 2.2 *Waste and amendments*

Food waste used in the study was fortified according to the standard coupon of food waste in Japan (as shown in Table 1), with a small change of that no egg shell was used because it was supposed to be not involved in the transformation processes of carbon and nitrogen. Chicken bone,

fish and rice were cooked before using. After being ground by a disposer (Emerson Electric Co., USA), the materials were stored at 4°C in refrigerator. The waste obtained finally contains approximately 44.60% of carbon, 2.10% nitrogen and 0.02% phosphorus. In the study, different loading rates and application patterns were used. For daily application pattern, 490g food waste was added into reactor every day until the end of composting. For single application pattern, 3,920g food waste was applied in low-loading-rate treatments one time at the beginning and no additional waste was added (unless specially indicated) until the end of composting and 7,350g was added in high-loading-rate treatments.

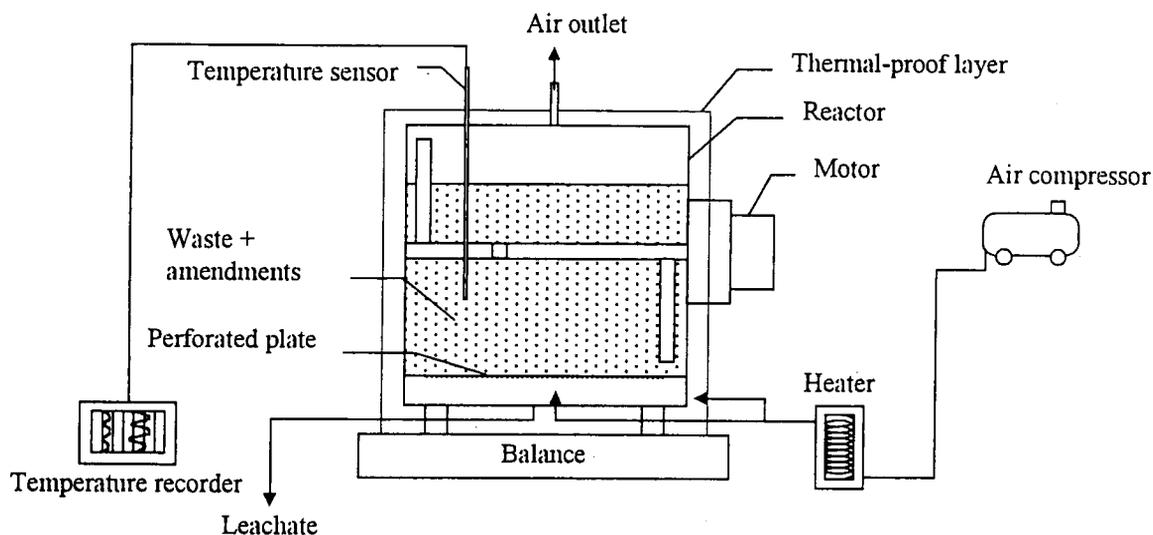


Fig. 1 Diagram of the experimental apparatus

Table 1. Composition of food waste (g·person<sup>-1</sup>·day<sup>-1</sup>)

Component	Chicken bone	Fish	Apple	Banana peel	Grape peel
Amount	20	25	25	25	25
Component	Cabbage	Carrot	Rice	Tea residue	Egg shell
Amount	45	45	25	10	5

Sawdust and compost were used as amendments to improve the air dispersion in waste pile and improve the degradation. To compare the effects of them on the greenhouse gas generation, different rates of sawdust and compost were used in various reactors. In all experimental runs, 1,500g sawdust was added. The sawdust we used, Biochip, was purchased from National Company (Japan). Beside its effect on air dispersion, Biochip also consists of necessary microorganisms for the biodecomposition.

### 2.3 Sampling and analysis

Air samples were collected from the air outlet at the top of reactor at different time interval to determine concentrations of CH<sub>4</sub> and N<sub>2</sub>O. N<sub>2</sub>O was analysed on gas chromatography (Shimadzu Co., Japan) implemented with electron capture detector and Poropak Q column, using 40 ml·min<sup>-1</sup> argon containing 5% methane as carrier gas, temperature of the detector and oven were 340°C and 80°C respectively. CH<sub>4</sub> was analysed on gas chromatography (Shimadzu Co., Japan) with flame ionic detector and Poropak Q Column, using nitrogen as carrier gas, and temperature of oven and injector port were 60°C and 80°C, respectively. O<sub>2</sub>, CO<sub>2</sub> and NH<sub>3</sub> were directly determined at the outlet with sensors (Gastec Co., Japan)

Waste was sampled from different corners of the reactor at different time intervals. To determine the contents of NH<sub>4</sub><sup>+</sup>, NO<sub>3</sub><sup>-</sup>-N, NO<sub>2</sub><sup>-</sup>-N and TN, approximately 2 g of waste was extracted by 20 ml 2M KCl solution for 30 min on a shaker. After being centrifuged at 4,000rpm for 10 min,

the supernatant was filtered through glass fiber filter (0.4 $\mu$ m). Concentrations of nitrogen ions and TP in the obtained solution were finally analyzed by automated colorimetric methods on a TRAACS-800 instrument.

### 3. RESULTS AND DISCUSSION

#### 3.1 Reproducibility of reactor performance

The reproducibility of the experimental aerated composting system can be assessed by the analysis of oxygen and temperature profiles from two experiments with identical initial waste amendments, adding pattern and inlet air condition. In the first experiment, in which only sawdust was used and daily addition pattern of food waste (DAP, hereafter) was employed, both of the duplicates showed a 40–46°C temperature peak between the 6<sup>th</sup> day and a stabilizing temperature profile after the 10<sup>th</sup> day. Oxygen levels in the exhausted air decreased firstly to 15% and then slowly recovered to around 19% at the 11<sup>th</sup> day. In the second experiment, in which compost was added in addition to sawdust and single addition pattern of food waste (SAP, hereafter) was followed. Temperature peaked at the fifth day to 42–44°C and slowly decreased to around 32°C after 10 days for both trials. Oxygen percentage in the exhausted air decreased more sharply than that in treatment without compost, to 13% and recovered to 19% at the 11<sup>th</sup> day. Within our experimental period, the average temperature difference between the two similar trials in each treatment group was less than 2°C. Similarly, the average difference between oxygen profiles for treatments with identical operation condition were less than 0.5%. Therefore, the analysis of the differences between replicate profiles of temperature and oxygen showed the test system was able to duplicate well within our experiment period.

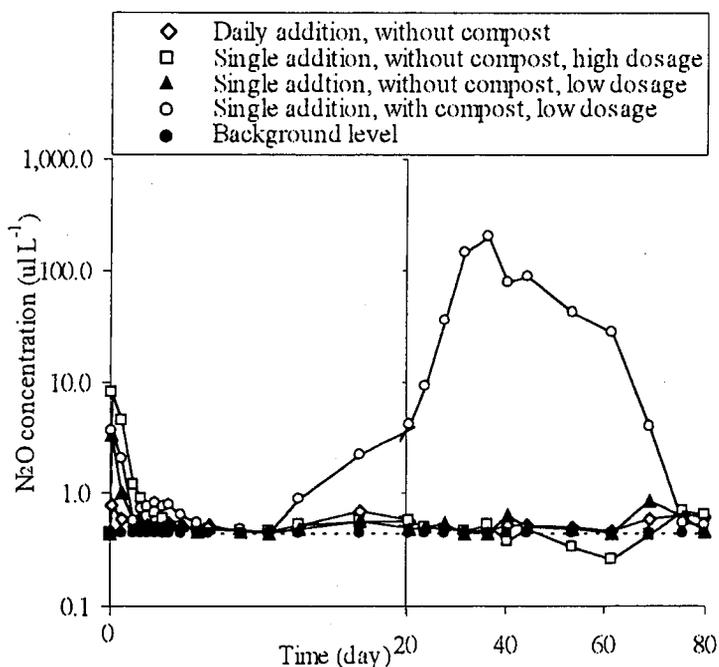


Fig. 2 Nitrous oxide emission from aerated composting system

#### 3.1 Emissions of $N_2O$ and $CH_4$

Fluctuation of  $N_2O$  concentration at the air outlet of reactor during the aerated composting of food waste is shown in Fig. 2. During the first day, a small peak of  $N_2O$  emission was observed in all treatment groups. The concentration of  $N_2O$  at the outlet of reactor increased sharply to 0.79, 3.32 and 8.11  $ul \cdot L^{-1}$  for DAP, SAP-low dosage and SAP-high dosage treatments, respectively, suggesting good correlation between  $N_2O$  generation and the amount of waste in each reactor. After two days,  $N_2O$  concentrations in all non-compost treatments decreased to 0.53  $ul \cdot L^{-1}$  in average, slightly higher than the background level of 0.45  $ul \cdot L^{-1}$  in atmosphere. The general emission rate of  $N_2O$  was 2.4  $ul$  per hour if the waste addition shock was not considered.

In treatments applied with compost, two peaks were observed for  $N_2O$  emission (Fig. 2). The first peak appeared simultaneously with that in non-compost treatment, with an increment of 1  $ul \cdot L^{-1}$  at the emission peak. The second emission peaks were detected after the 12<sup>th</sup> day in reactors applied with compost. Nitrous oxide concentrations in the exhausted gas increased steadily to 202.67  $ul \cdot L^{-1}$  at the 36–40<sup>th</sup> day of incubation and slowly returned to the background level until the end of experiment, while no such a peak was found in non-compost treatments.

No methane production was detected in reactors without compost throughout the experiment (Fig. 3). Methane concentrations in the exhausted gases from these reactors were near to the background level ( $2.06 \text{ ul}\cdot\text{L}^{-1}$  in the vicinity of experimental chamber) all the time even though large amount of waste was added in reactors with SAP at the beginning of experiment. However, high concentrations of methane were detected in the reactors applied with compost 2 days after the commencement of operation. Methane production increased steadily and peaked 20 days after the commencement of composting. This result indicated the presence of anoxic or anaerobic microenvironment in waste during the operation of aerated composting system.

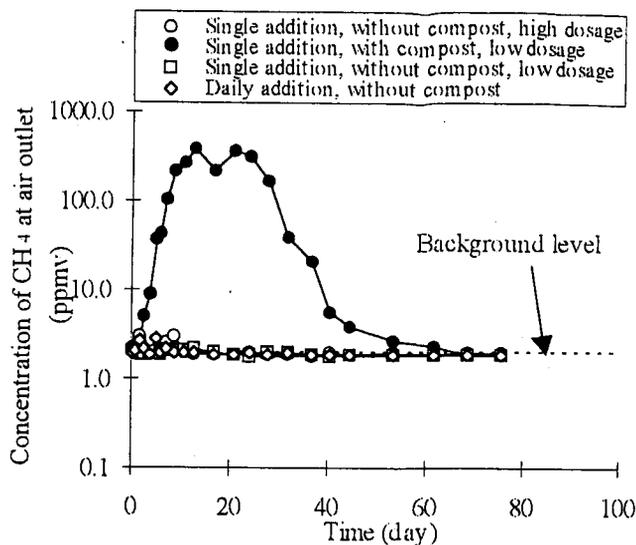


Fig. 3 Methane production from aerated composting system

concentration in the exhausted gas and ammonia, nitrate and total nitrogen content in waste. For nitrite, however, different results were obtained for the compost-applied and non-compost treatments.  $\text{N}_2\text{O}$  emission was found to linearly correlated with nitrite content in waste in the compost-applied treatments (Fig. 4) while no significant correlation was observed in non-compost treatment. This phenomenon was further certified by the immediate increase of  $\text{N}_2\text{O}$  emission after a nitrite shock, by adding sodium nitrite into waste mixture (Fig. 5). In previous studies, similar stimulation of  $\text{N}_2\text{O}$  production by increased  $\text{NO}_2^-$  was found in aerobic wastewater treatment

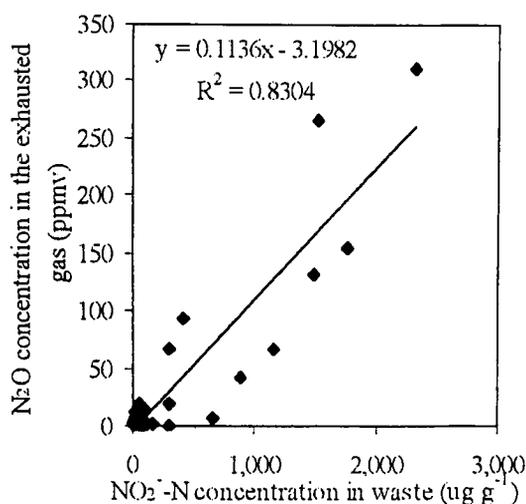


Fig. 4 Correlation between  $\text{N}_2\text{O}$  emission and nitrite content in waste mixture

### 3.2 Relations between $\text{N}_2\text{O}$ emission and nitrogen forms in waste mixture

Previous studies showed that  $\text{N}_2\text{O}$  production was closely related to the nitrogen forms and amount. Addition of ammonia, nitrate or other forms of nitrogen might result in substantial increase in  $\text{N}_2\text{O}$  emissions from manure land application (Cates and Keeney, 1987; Burford and Bremner, 1975; Boustany et al., 1997) and composting system (Mahimairaja, 1995). In our study, ammonia, nitrate, nitrite and total nitrogen were determined at the time of each flux measurement. No significant statistical correlation was observed between  $\text{N}_2\text{O}$

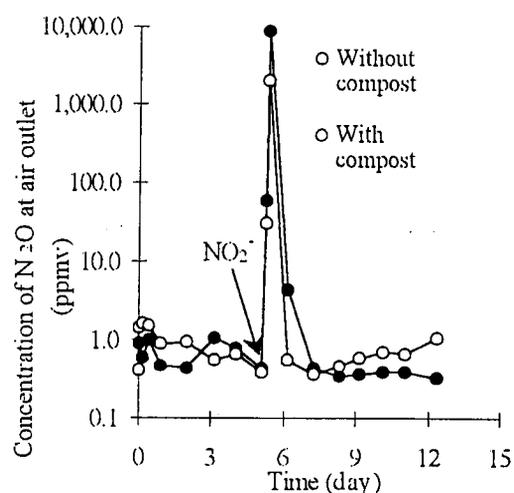


Fig. 5 Effect of nitrite shock on  $\text{N}_2\text{O}$  emission during aerated composting

process like active sludge, nitrification (Schulthess et al., 1994, 1995; Wild et al., 1994, 1995 and Zheng et al., 1994). The good correlation between  $\text{N}_2\text{O}$  emission and  $\text{NO}_2^-$  shows the dependence of  $\text{N}_2\text{O}$  production on  $\text{NO}_2^-$  level, and suggests the presence of a  $\text{N}_2\text{O}$  generation pathway from  $\text{NO}_2^-$  to

$N_2O$  in the compost-applied treatments. This reduction step can be fulfilled by either nitrifier or denitrifier.

In Fig. 5, nitrite shock also caused an increase in  $N_2O$  generation in non-compost treatments, suggesting the existence of similar  $N_2O$  generation mechanism. However, during the composting process, no evident emission was detected in non-compost treatments after the 8<sup>th</sup> day. In waste mixture, no nitrite accumulation was detected, therefore, lack of nitrite resulted into the low  $N_2O$  emission in these treatments. Meanwhile significantly higher ammonia emission was found in non-compost treatments (Fig. 6), indicating that large amount of nitrogen was lost through ammonia volatilization.

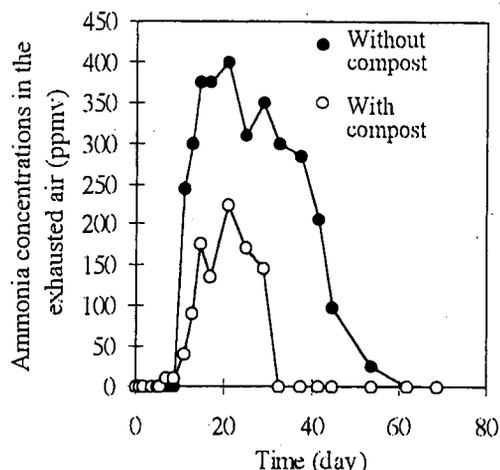


Fig. 6 Ammonia volatilization during aerated composting

### 3.3 Effect of carbon content on $N_2O$ production

Previous studies showed that carbon availability also influence  $N_2O$  production in wastewater treatment system. Improper rates between organic carbon and nitrogen was found to increase  $N_2O$  emission (Inamori et al., 1994; Hanaki et al., 1990). In our study, no significant correlation was found between  $N_2O$  emission and the rate of dissolved organic carbon and nitrogen (DOC/TN) for both compost-applied and non-compost treatments. Nevertheless, it was apparent that the higher emission of  $N_2O$  ( $>10 \text{ ul}\cdot\text{L}^{-1}$ ) all occurred while DOC/TN ratio was lower than 5 (Fig. 7). In another experiment, besides food waste added in compost-applied treatments at the beginning, same aliquot of food waste was re-added into reactor for two times at the 5<sup>th</sup> and 14<sup>th</sup> day, respectively, together with same ratio of sawdust and compost. As a result, the second emission peak appeared in compost-applied treatments was postponed for 33 days (Fig. 8). The reason for this is that re-addition of food waste cause prolonging of ammonification and delaying of other steps of nitrogen transformation. Our result implies that in

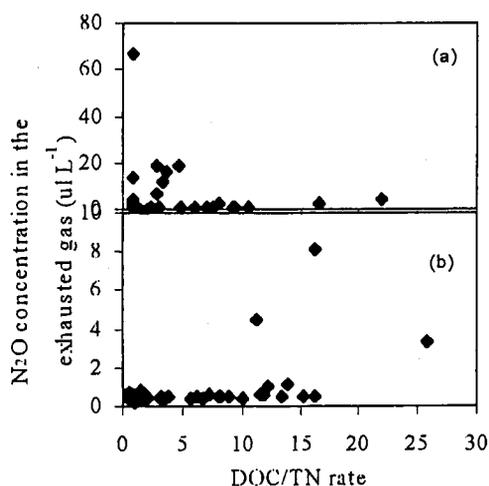


Fig. 7 Relation between  $N_2O$  emission and DOC/N rate in waste mixture; (a) with compost; (b) without compost

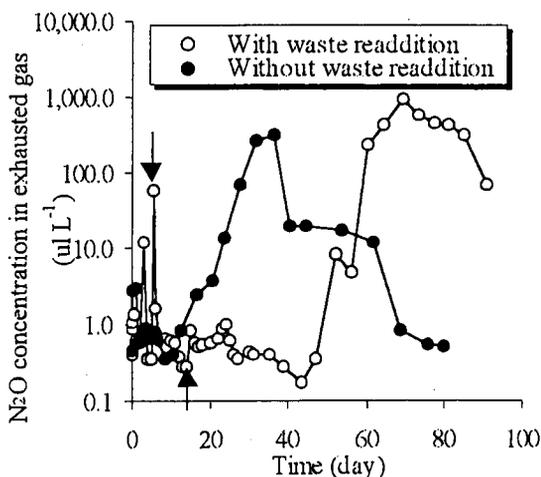


Fig. 8 Effect of waste re-addition on  $N_2O$  emission; arrows show the waste re-addition point

practical composting system, in which daily application pattern or multi-application pattern (waste is added one time in several days) were used,  $N_2O$  emission may not be evident because of the above “postponing effect”, but there is still potentiality for  $N_2O$  emission in composting product. This will possibly result into large  $N_2O$  production after the land application of composting product.

### 3.4 Effect of aeration rate on emission of $N_2O$ and $CH_4$

In aerated composting treatment, aeration is of importance not only for the treatment performance and odor control, but also for greenhouse gas emission. As shown in Fig. 9, when the aeration rate is increased from 0.70 to 1.40 and 2.10  $\text{L}\cdot\text{min}^{-1}\cdot\text{kg}^{-1}$  (dry weight of food waste),  $\text{N}_2\text{O}$  production increase significantly. The increment is more obvious from 1.40 to 2.10  $\text{L}\cdot\text{min}^{-1}\cdot\text{kg}^{-1}$ . With increasing aeration rate, air-stripping effect become stronger, nitrous oxide formed in compost pile is quickly stripped away before being oxidized further. This is thought to be the main reason for high  $\text{N}_2\text{O}$  generation in treatments with high aeration rates. When the aeration rate is reduced from 0.70 to 0.56 and 0.14  $\text{L}\cdot\text{min}^{-1}\cdot\text{kg}^{-1}$  respectively in treatments applied with compost, the second  $\text{N}_2\text{O}$  emission peak disappear (Fig. 10); therefore, the total  $\text{N}_2\text{O}$  production was greatly reduced. In

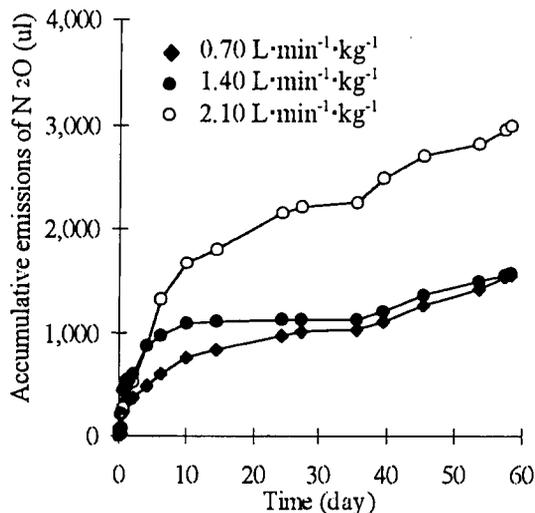


Fig. 9 Effect of aeration rate on  $\text{N}_2\text{O}$  emission from treatments without compost

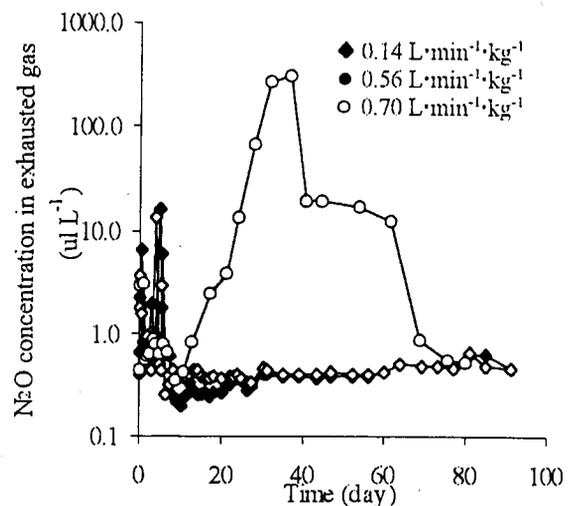


Fig. 10 Effect of aeration rate on  $\text{N}_2\text{O}$  emission from treatments with compost

these treatments, ammonia volatilization was found to be larger and remove large amount of nitrogen before the parent chemical for  $\text{N}_2\text{O}$  being formed. The result suggests that using low aeration rate might be an approach to control  $\text{N}_2\text{O}$  emission. However, the aeration rate reduction is limited because too low concentration of oxygen will result in anaerobic environment and enlarge denitrification process, which has already been known to generate large amount of  $\text{N}_2\text{O}$  (van Kessel et al., 1993).

Moreover, methane generation was found to increase in treatments with low aeration rates. It indicates that a balance point of aeration rate need to be determined in future studies with the consideration of all greenhouse gases rather than the individual gas.

### 3.5 Possible mechanisms for $\text{N}_2\text{O}$ and $\text{CH}_4$ emissions

In the non-compost treatments, two mechanisms were considered to be responsible for the  $\text{N}_2\text{O}$  emission peak at the beginning. One is the possible imbalance of various microbial species in the Biochip. As it is commonly known, various microorganisms conduct the different steps of the oxidation of ammonia and nitrite during nitrification, which is the main procedure in our system. (Williams et al., 1992). The imbalance between species in charge of different nitrification steps in Biochip might cause a residue of the intermediate product, like  $\text{N}_2\text{O}$  and  $\text{NO}_2^-$ . Another reason for the high  $\text{N}_2\text{O}$  emission at the beginning might be the metabolism of the microbial community coming from food waste. In a preliminary experiment, food waste was found to generate  $\text{N}_2\text{O}$  even stored at  $4^\circ\text{C}$ , suggesting the existence of  $\text{N}_2\text{O}$  generation mechanism in food waste. These "aboriginal" microorganisms may survive at the beginning of experiment when the temperature was relatively low and contribute to the  $\text{N}_2\text{O}$  peak. With the increase of temperature, these microorganisms perished or were outnumbered by the thermophilic species.

In compost-applied treatments,  $N_2O$  emission shortly after the commencement was higher than non-compost trials. With the consideration of the previously reported high  $N_2O$  emission from composted manure (Kuroda et al., 1996; Mahimairaja et al., 1995), it is not surprising to observe the increased generation of  $N_2O$  during this period. The addition of  $N_2O$  producer by compost amendment may increase the  $N_2O$  generation rate. This effect, together with the activity of "aboriginal" microorganism in food waste and residue of metabolism intermediates caused by imbalance of microbial community in Biochip, forms the first  $N_2O$  emission culmination.

Within our experimental period, the second emission peak was found only in the compost-applied treatments. As a result, accumulative  $N_2O$  emissions surpassed that from non-compost treatment, even more than 2 times amount of waste was involved (Fig. 2). The result implies that there was other  $N_2O$  generation procedures in addition to those introduced above, which overshadowed the effect of waste amount. With the consideration of good correlation between  $N_2O$  and  $NO_2^-$  during the later period of compost-applied composting, it seems that  $N_2O$  was reduced from  $NO_2^-$ , however, this transformation reaction usually occurs at anoxic or anaerobic conditions. In our study, methane was detected simultaneously in compost-applied group (Fig. 3) implying existence of low-oxygen-level environment. Though aeration was employed, anoxic or anaerobic microenvironment may still exist inside the waste particles, ranging from several  $\mu m$  to 4-5mm in diameter in this experiment. Similar micro-environmental effects had been reported by previous studies in aerobic soil (Goodroad and Keeney, 1984; Parkin, 1987). Therefore the possible existence of nitrite reduction, caused by either nitrifiers or denitrifiers in the microsite, might contribute to the emission peak of  $N_2O$  at the 36-40<sup>th</sup> day, in addition to nitrification which was certified to exist in our system by the sequential production of carbon dioxide, ammonia, nitrite and nitrate. Coexistence of nitrification and denitrification has been previously reported in both soil profile (Goodroad and Keeney, 1984; Parkin, 1987) and wastewater treatment (Schulthess et al., 1994, 1995; Wild et al., 1994, 1995 and Zheng et al., 1994).

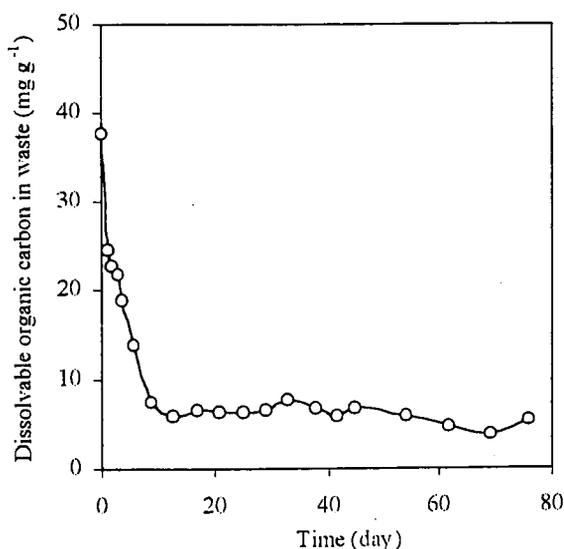


Fig. 12 Change of dissolved organic carbon with time in compost-applied treatments

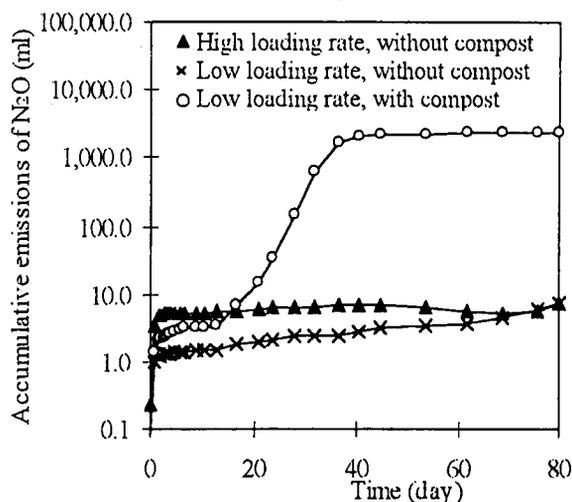


Fig. 11 Cumulative  $N_2O$  emission in compost-applied and non-compost treatments

If denitrifiers conducted denitrification caused the second  $N_2O$  emission peak,  $N_2O$  production should be proportion to the available carbon, as commonly identified before (Burford and Bremner, 1975; Cabrera et al., 1994) because denitrifiers are mostly heterotrophs relying on organic carbon to retrieve energy. However in Fig. 2,  $N_2O$  emission began to increase at the 12<sup>th</sup> day, shortly after the completion of available organic carbon (Fig. 12). Active metabolism of denitrifier was inconsistent with this result; therefore, nitrifiers were thought to be the main microorganisms that reduce  $NO_2^-$  to  $N_2O$ .

For lack of information on the composition of microbial community involved, it is difficult in this article to directly specify which microorganism species play the central role in  $N_2O$  generation; however, the microorganism(s) causing rapid nitrite accumulation surely came from compost. In

non-compost treatments, though the anoxic microenvironment may also exist, significant N<sub>2</sub>O emission was not observed for the absence of nitrite accumulation.

#### 4. CONCLUSIONS

In this study, emissions of N<sub>2</sub>O and CH<sub>4</sub> from the aerated composting system were investigated. In non-compost treatments, one-peak emission curves were obtained. N<sub>2</sub>O was mainly generated at the beginning of composting and peaked during the first day. 2 days after, N<sub>2</sub>O emission decreases to 0.53 ul·L<sup>-1</sup> in average, near to the background level in the atmosphere (0.45 ul·L<sup>-1</sup>). In the compost-applied treatment, however, two-peak emission curves were observed. Besides the peak at the beginning, high emission occurred again in the later period and culminated around the 36-40<sup>th</sup> day. The cumulative emission from the 12<sup>th</sup> day to the end accounts for most of total N<sub>2</sub>O generation. In compost-applied treatment, good correlation between N<sub>2</sub>O concentration at the air outlet and NO<sub>2</sub><sup>-</sup> concentration in waste was observed. The relationship was further proved by a immediate increase in N<sub>2</sub>O emission after nitrite was added into waste mixture, indicating a generation pathway for N<sub>2</sub>O from NO<sub>2</sub><sup>-</sup> to N<sub>2</sub>O. The transformation step might be finished by nitrifiers.

Increasing aeration rate may increase N<sub>2</sub>O production because of the strong air-stripping effect. More than 80% of total N<sub>2</sub>O emission in compost-applied treatments can be removed while the aeration rate was reduced to 0.14L·min<sup>-1</sup>·kg<sup>-1</sup>; however, methane production increased significantly and offset effect of N<sub>2</sub>O reduction. Meanwhile, high ammonia volatilization at low aeration rate causes serious odor problem.

Methane was only detected in the compost-applied treatment, the high emission of it illustrates the involvement of anoxic/anaerobic microorganisms with the addition of the composted manure. The result suggests the existence of anoxic or anaerobic environment even though ventilation was employed.

#### 5. REFERENCE

1. Åkesson M, Nilsson P. Material dependence of methane production rates in landfills. *Waste Manage & Res* 1998; 16(2): 108-118.
2. Boltze U, de Freitas MH. Monitoring gas emissions from landfill sites *Waste Manage & Res* 1997; 15: 463-476.
3. Börjesson G, Svensson BoH. Seasonal and diurnal methane emissions from a landfill and their regulation by methane oxidation. *Waste Manage & Res* 1997; 15(1): 33-54.
4. Boustany RG, Crozier CR, Rybczyk JM, Twilley RR. Denitrification in a south Louisiana wet forest receiving treated sewage effluent. *Wetlands Ecol Manage* 1997; 4: 273-283.
5. Burford JR, Bremner JM. Relationships between the denitrification capacities of soils and total, water-soluble and readily decomposable soil organic matter. *Soil Biol Biochem* 1975; 7: 389-394.
6. Cabrera M, Chiang S, Merka W, Pancorbo O, Thompson S. Nitrous oxide and carbon dioxide emissions from pelletized and non pelletized poultry litter in incorporated into soil. *Plant Soil* 1994; 163: 189-196.
7. Cates Jr RL, Keeney D.R. Nitrous oxide production throughout the year from fertilized and manured maize fields. *J Environ Qual* 1987; 16(4): 443-447.
8. Czepiel P, Douglas E, Harriss R, Crill P. Measurements of N<sub>2</sub>O from composted organic wastes. *Environ Sci Technol* 1996; 30(8): 2519-2525.
9. Frette L, Gejlsbjerg B, Westermann P. Aerobic denitrifiers isolated from an alternating activated sludge system. *FEMS Microbiol Ecol* 1997; 24: 363-370.
10. Goodroad LL, Keeney DR. Nitrous oxide production in aerobic soils under varying pH, temperature and water content. *Soil Biol Biochem* 1984; 16(1): 39-43.
11. Hanaki K, Wantawin C, Gujer W. Effects of the activity of heterotrophs on nitrification in a

- suspended-growth reactor. *Wat Res* 1990; 24: 289-296.
12. Inamori Y, Hosomi M, Sudo R. Greenhouse effect gas control producing from wastewater treatment processes. *J Wat Waste* 1991; 34(1): 28-34 (in Japanese).
  13. Inamori Y, Mizuochi M, Terunuma H, Yamamoto M, Makuta T, Utida T, Kimochi Y. Effects of anaerobic condition on biological nitrogen, phosphorus removal and inhibition of greenhouse effect gas by intermittently aerated activated sludge process. In: *Proceedings of the 28<sup>th</sup> Annual Symposium of Japan Society on Water Environment*, 1994; 278-279 (in Japanese).
  14. Kjeldsen P, Fischer EV. Landfill gas migration-field investigations at Skellingsten landfill, Denmark. *Waste Manage & Res* 1995; 13: 467-484.
  15. Kuroda K, Osada T, Yonaga M, Kanematu A, Nitta T, Mouri S, Kojima T. Emissions of malodorous compounds and greenhouse gases from com composting swine feces. *Bioresource Technol* 1996; 56: 265-271.
  16. Liu AH, Kunz CO. Gas-flow model to determine methane production at sanitary landfills. *Environ Sci Technol* 1981; 15(4): 436-440.
  17. Mahimairaja S, Bolan NS, Hedley MJ. Denitrification losses of N from fresh and composted manures. *Soil Biol Biochem* 1995; 27(9): 1223-1225.
  18. Parkin TB. Soil microsites as a source of denitrification variability. *Soil Sci Soc Am J* 1987; 51: 1194-1199.
  19. Paul JW, Beauchamp EG, Zhang C. Nitrous and nitric oxide emissions during nitrification and denitrification from manure-amended soil in the laboratory. *Can J Soil Sci* 1993; 73: 539-553.
  20. Schulthess RV, Wild D, Gujer W. Nitric and nitrous oxides from denitrifying activated sludge at low oxygen concentration. *Wat Sci Technol* 1994; 30: 123-132.
  21. Schulthess RV, Kuhn M, Gujer W. Release of nitric and nitrous oxides from denitrifying activated sludge. *Wat Res* 1995; 29: 215-226.
  22. Tanaka M, Miyazaki M, Watanabe I. CH<sub>4</sub> and N<sub>2</sub>O emission from waste disposal facilities in Japan. In: Minami K, Mosier A, Sass R, editors. *CH<sub>4</sub> and N<sub>2</sub>O Global Emissions and Controls from Rice Fields and Other Agricultural and Industrial Sources*, Yokendo Publishers, Tokyo. 1994, pp. 181-186.
  23. Tanikawa N, Imai T, Urano K. Characteristics of continuous analyzers for nitrous oxide in flue gas from municipal incinerators. *Sci Total Environ* 1995; 175: 189-198.
  24. Tsujimoto Y, Masuda J, Fukuya J, Ito H. N<sub>2</sub>O emissions at solid waste disposal sites in Osaka city. *J Air & Waste Manage Assoc* 1994; 44: 1313-1314.
  25. van Kessel C, Pennock DJ and Farrell RE. Seasonal variations in denitrification and nitrous oxide evolution at the landscape scale. *Soil Sci Soc Am J* 1993; 57: 988-995.
  26. Wang YS, Odle III WS, Eleazer WE, Barlaz MA. Methane potential of food waste and anaerobic toxicity of leachate produced during food waste decomposition. *Waste Manage Res* 1997; 15: 149-167.
  27. Wild D, Schulthess RV, Gujer W. Structure modeling of denitrification intermediated. *Wat Sci Technol* 1995; 31: 45-54.
  28. Wild D, Schulthess RV, Gujer W. Synthesis of denitrification enzymes in activated sludge: modeling with structure biomass. *Wat Sci Technol* 1994; 30: 113-122.
  29. Williams EJ, Hutchinson GL, Fehsenfeld FC. NO<sub>x</sub> and N<sub>2</sub>O emissions from soil. *Global Biogeochem Cycles* 1992; 6(4): 351-388.
  30. Wu XL, Kong HN, Mizuochi M, Inamori Y, Huang X, Qian Y. Nitrous oxide emission from microorganisms. *Japanese J Wat Treatment Biol* 1995; 31(3): 151-160.
  31. Zheng H, Hanaki K, Matsuo T. Production of nitrous oxide gas during nitrification of wastewater. *Wat Sci Technol* 1994; 30(6): 134-141.